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► To cite this version:

M. Zavala, S. C. Herndon, E. C. Wood, J. T. Jayne, D. D. Nelson, et al.. Comparison of emission ratios from on-road sources using a mobile laboratory under various driving and operational sampling modes. *Atmospheric Chemistry and Physics Discussions*, 2008, 8 (2), pp.8059-8090. hal-00304131

HAL Id: hal-00304131

<https://hal.science/hal-00304131>

Submitted on 23 Apr 2008

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Comparison of emission ratios from on-road sources using a mobile laboratory under various driving and operational sampling modes

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8, 8059–8090, 2008

**Variability of mobile
emissions sampled
using a mobile lab**

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



Received: 26 March 2008 – Accepted: 31 March 2008 – Published: 23 April 2008

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Published by Copernicus Publications on behalf of the European Geosciences Union.

ACPD

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Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

Abstract

Mobile sources produce a significant fraction of the total anthropogenic emissions burden in large cities and have harmful effects on air quality at multiple spatial scales. Mobile emissions are intrinsically difficult to estimate due to the large number of parameters affecting the emissions variability within and across vehicles types. The MCMA-2003 Campaign in Mexico City has showed the utility of using a mobile laboratory to sample and characterize specific classes of motor vehicles to better quantify their emissions characteristics as a function of their driving cycles. The technique clearly identifies “high emitter” vehicles via individual exhaust plumes, and also provides fleet average emission rates. We have applied this technique to Mexicali during the Border Ozone Reduction and Air Quality Improvement Program for the Mexicali-Imperial Valley in 2005. In this paper we analyze the variability of measured emission ratios for emitted NO_x , CO, specific VOCs, NH_3 , and some primary fine particle components and properties obtained during the Border Ozone Reduction and Air Quality Improvement Program for the Mexicali-Imperial Valley in 2005 by deploying a mobile laboratory in roadside stationary sampling, chase and fleet average operational sampling modes. The measurements reflect various driving modes characteristic of the urban fleets. The observed variability for all measured gases and particle emission ratios is greater for the chase and roadside stationary sampling than for fleet average measurements. The fleet average sampling mode captured the effects of traffic conditions on the measured on-road emission ratios, allowing the use of fuel-based emission ratios to assess the validity of traditional “bottom-up” emissions inventories. Using the measured on-road emission ratios, we estimate CO and NO_x mobile emissions of 175 ± 62 and 10.4 ± 1.3 metric tons/day, respectively, for the gasoline vehicle fleet in Mexicali. Comparisons with similar on-road emissions data from Mexico City indicated that fleet average NO emission ratios were around 20% higher in Mexicali than in Mexico City whereas HCHO and NH_3 emission ratios were higher by a factor of 2 in Mexico City than in Mexicali. Acetaldehyde emission ratios did not differ significantly whereas selected aromatics

ACPD

8, 8059–8090, 2008

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



VOCs emissions were similar or smaller in Mexicali. On-road heavy-duty diesel truck (HDDT) nitrogen oxides emissions were measured near Austin, Texas, as well as in both Mexican cities, with NO_y emission ratios in Austin<Mexico City<Mexicali.

1 Introduction

5 Emissions from transportation sources, primarily on-road motor vehicles, are generally the largest contributors to criteria air pollutants such as CO, NO_x, and selected volatile organic compounds (VOCs) in urban areas; On-road vehicles are also major sources of fine primary particle emissions and specific air toxics (Molina et al., 2004). Despite their importance in determining air quality levels, the estimation of mobile emission
10 sources is challenging because multiple parameters affect the variability of on-road mobile emissions within and across vehicles types (Cadle et al., 2007). Factors such as engine size and type, fuel composition, temperature and pressure are directly linked to the combustion efficiency (and therefore the emission rates) of in-use vehicles; other external factors such as driving cycles and the character and maintenance of fuel deliv-
15 ery and emission control systems also decisively affect the variability and composition of mobile emissions (NARSTO, 2005).

All these vehicle parameters and driving conditions significantly affect the observed variability of on-road emissions for a given vehicle type. Because the vehicle fleet in an urban area is composed of a large number of vehicle types, “fleet-average” emis-
20 sion characteristics have in fact an associated intrinsic variability. In this work, “fleet-average” describes conditions where individual plume emissions from a large number of vehicles are captured during sampling; the longer the sample data extends, the more probable the overall emission characteristics of the fleet are captured. The observed variability during the measurement of on-road emissions in fleet average driving con-
25 ditions is the result of the individual emission variability from a wide range of sampled vehicles. As a result, point estimates (e.g. average emissions of a given pollutant) of the rate of mobile emissions in an urban area are of limited value unless a description

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



of their associated variability is available.

Additionally, cross-validation and inter-comparisons of mobile emission measurements using different emission measurement techniques (such as remote sensing, mobile laboratories, dynamometer and tunnel studies) are intrinsically difficult to perform due to differences in sampling times and frequencies, pollutant measurements instrumentation, sample size and analysis assumptions for each measurement. When using a mobile laboratory for sampling on-road emissions several operational sampling modes can be used. This includes: roadside stationary sampling with wind advection bringing plumes to the sample location and 'on-road' sampling chase and fleet average measurements. Due to differences in sampling times, sample sizes and frequencies, each of these sampling conditions capture some portion of the actual variability.

During the MCMA-2003 Campaign, the Aerodyne Research Inc. (ARI) mobile laboratory was deployed in the Mexico City Metropolitan Area (MCMA) to sample and characterize specific classes of motor vehicles to better quantify their emissions characteristics as a function of their driving cycles (Molina et al., 2007). Emission ratios for NO_x , NO_y , NH_3 , H_2CO , CH_3CHO , and other selected volatile organic compounds (VOCs) were estimated for chase sampled vehicles in the form of frequency distributions and for the fleet averaged emissions (Zavala et al., 2006). The results indicate that the technique is capable of differentiating among vehicle categories and fuel type under real world driving conditions. We extended this technique to Mexicali, Baja California, Mexico during the Border Ozone Reduction and Air Quality Improvement Program for the Mexicali-Imperial Valley in 2005 (Mendoza et al., 2007).

This paper discusses the measurements of on-road mobile emissions obtained from April 12-23, 2005 in Mexicali during the field campaign under different driving and operational sampling modes using the ARI mobile laboratory. The driving modes represented various speed and congestion characteristics of the sampled fleet. Sampling modes included: 1) roadside stationary sampling of individual identified vehicle emission plumes, 2) "chase" experiments where the mobile laboratory followed a specific vehicle for several minutes repeatedly sampling its exhaust plume, and 3) on-road fleet

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



average sampling modes where no attempt is made to distinguish plumes from individual vehicles and all intercepted vehicle emissions plumes are counted and weighted equally.

We present a comparison of the on-road emission measurements in Mexicali with corresponding measurements obtained during the MCMA-2003 field campaign (Molina et al., 2007). This constitutes a unique opportunity to compare the vehicle fleet emission characteristics of a megacity and a smaller urban area in the same developing country. Since the measurements were obtained using the same technique, assumptions and instrumentation in the two campaigns, the observed differences in the comparison are more likely to be the result of actual differences in fleet emission characteristics between the two cities. This direct comparison is useful for understanding the fast-evolving characteristics of the vehicle fleet in a US-Mexican border city. In addition, during 8–9 May 2003 the ARI mobile laboratory obtained on-road measurements of heavy-duty diesel trucks (HDDTs) in and near Austin, TX in order to capture Mexican and US HDDTs emissions using the chase technique. We also compare the results obtained in Austin, with measurements obtained from individual HDDTs in Mexico City and Mexicali.

2 Methodology

The mobile laboratory deployed during the Mexicali field campaign was equipped with several high time resolution and high sensitivity instruments as described in detail in Kolb et al. (2004), Herndon et al. (2005), and Zavala et al. (2006). These included Tunable Infrared Laser Differential Absorption Spectrometers (TILDAS) for measuring selected gaseous pollutants, a Proton Transfer Reaction Mass Spectrometry (PTR-MS) for measuring selected VOCs, a commercial NO/NO_y chemiluminescent detector modified for fast response measurements, and a Licor Non-Dispersive Infrared (NDIR) instrument for CO₂. An Aerosol Mass Spectrometer (AMS), a Condensation Particle Counter (CPC) and a Multi Angle Absorption Photometer (MAAP) were also deployed

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



to retrieve information regarding the composition, number and light absorbing carbon information of emitted particles. High time resolution instrumentation allows the mobile laboratory to capture the temporal pollutant concentrations variability of the turbulent exhaust plumes as they are dispersed into the surrounding air. Other instruments on board the mobile laboratory included a Global Positioning System (GPS), a sonic anemometer and a video camera used to obtain target vehicle information. The mobile laboratory's velocity and acceleration were measured and recorded continuously to characterize the driving mode conditions during the sampling. Local atmospheric parameters including pressure, temperature, and relative humidity were also measured continuously.

For the Mexicali study, a total of 98 valid mobile emission ratio experimental periods were obtained during the analysis of 14.5 hours of on-road and roadside data. The samples comprised a variety of driving modes (e.g., idling, acceleration, cruising, etc.), fuel types (gasoline and diesel), vehicle model years, and vehicle types (light-duty gasoline vehicles (LDGVs) and HDDTs). Three types of operational sampling modes were used to obtain on-road vehicle emission data: 1) individual identified vehicle emission plumes measured by roadside stationary sampling; in this mode, emission ratios from individual emission plumes were obtained during periods of stationary sampling along the road whenever the wind was favourable for transporting the vehicle's emissions to the mobile laboratory sampling port, 2) chase experiments where the mobile laboratory followed specific vehicles, primarily heavy-duty diesel trucks and buses, repeatedly sampling their exhaust plumes for several minutes, and 3) on-road fleet average sampling modes where no attempt was made to distinguish plumes from individual vehicles and all intercepted vehicle emissions plumes from both passing and oncoming vehicles are counted and weighted equally; in this mode, emission ratios were obtained by analyzing the periods in which the emission signatures from surrounding vehicles were sufficiently mixed by the time they were sampled by the mobile lab.

The specific analytical procedures for obtaining the emission ratios in the aforementioned operational sampling modes have been described by Zavala et al. (2006).

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



Approximately 55 individual vehicles were characterized by roadside sampling, 19 by dedicated on-road chase, and 24 fleet average experiments sampled between a few tens to several hundred vehicles. All identified exhaust pollutant species are correlated with the excess (above background) CO₂ concentration, a tracer of combustion, allowing molar emission ratios to be computed for each measured exhaust pollutant. Fuel based emission indices (gram of pollutant to liter of fuel consumed) can readily be computed using the fuel properties from the observed molar emission ratios (Herndon et al., 2004b).

3 Results

3.1 Roadside stationary sampling plumes

For approximately 1.5 h on 22 April 2005, the mobile laboratory obtained on-road measurements of emission ratios in stationary sampling mode by situating on the side of a one-way road with moderate traffic and sampling dozens of individual plumes from passing vehicles. The road had no visible grade and was surrounded by open fields. Sampled vehicles included both LDGVs and HDDTs travelling from moderate to high speed. Other measurements of individual vehicle emission plumes were obtained during shorter periods of stationary road-side sampling during the campaign. These will not be presented in this section but are included for the inter-comparison with other sampling operational modes. The mobile laboratory was situated in the prevailing downwind direction from the emitting vehicles as consistently as possible because, in this type of operational sampling mode, a successful measurement of an emission exhaust signature from a passing vehicle is highly dependant on the predominant wind direction and speed at the time of the exhaust event (Herndon et al., 2005).

Once a plume exhaust is emitted from the passing vehicle, the aerosol and gaseous exhaust components are rapidly decelerated in the surrounding ambient air. The initial exhaust also has a significantly higher temperature than the background air. The pres-

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



ence of advection and induced turbulence produces rapid dilution and cooling of the emitted exhaust, dominated by small eddies generated by the inertial wake left by the vehicle (Dong and Chang, 2006). As the temperature gradient between the plume and the surrounding air decreases, further dilution is controlled by the local wind advection and turbulence, and the emission plumes slowly approach to background on-road concentrations (Wang et al., 2006).

Given the relatively small mass flux exhaust intensity for some vehicles and the short distance between the vehicle's emission exhaust location and the laboratory's sampling port, the signatures of the exhausts plumes recorded by the instruments last typically only a few seconds before they are highly diluted by the background air. Therefore, a characteristic time window of only a few seconds exists for high signal to background exhaust emission measurements.

The real-time trace gas and fine particle matter (PM) instruments aboard the mobile laboratory can resolve these short duration plumes allowing successful measurements of individual vehicle emission plumes as long as the selected road was not too heavily travelled. In the cases analyzed, unequivocal distinction of emission signatures for individual passing vehicles was possible when a relatively large time elapsed between passing vehicles. Additionally, analysis of recorded wind direction and speed in conjunction with the video camera helped to identify specific vehicles that produced the detected plumes. Highly sensitive and high time-response instruments are clearly critical for obtaining emission ratios for this type of sampling.

Figure 1 shows an example of stationary sampling emission plumes of a LDGV and a HDDT. As shown in Fig. 1, the sampled plumes lasted from 10 to 20 s before the emission signature is indistinguishable from the background on-road air. The combustion signature of the plume is observed by the high correlation of the emitted pollutants to above background CO₂ concentrations. In the particular case of the LDGV shown in Fig. 1, the vehicle had very high concentrations of most emitted pollutants and consequently high emission ratios. However, there is a clear distinction between the emission ratios of the two types of vehicles sampled. The CO and VOCs sampled in the case

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[◀](#)[▶](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

of the HDDT are significantly lower than the LDGV whereas the emitted NO, particle number density and the organic PM component are of the same magnitude or higher. Also as noted in Fig. 1 is the fact that, except for the organic component, most of the non-refractory components of the aerosols sampled with the AMS had negligible or poor correlations with CO₂. This may be explained by the ambient secondary formation nature of the chloride, nitrate and ammonium aerosol components, the short time between the emission and the sampling, and the short duration of the sampling time for the plume. The difference of the peak minus the background for the organic component is higher for the HDDT but is clearly significant in the LDGV as well, an indication of a high emitter vehicle.

Figure 2 shows a comparison of observed emission ratios of CO, NO, aromatic VOCs (considered here as the sum of benzene, toluene, C2benzenes and C3benzenes) and fine particle (10–1000 nm diameter) number density of emission plumes from individual gasoline and diesel vehicles sampled in roadside stationary mode. Each marker in the figure represents an individual measurement of an emission ratio for a given vehicle. Figure 2 demonstrates the co-emission nature of various pollutants for a given vehicle type and the variability between vehicle types. Figure 2 also indicates that the sampled LDGVs emitted higher aromatic and CO than the HDDTs, which is a direct result of the different combustion efficiencies for the two engine types. Similarly, within a given vehicle type, high CO and aromatic content in a vehicle's exhaust may be an indication of poor combustion efficiency, probably due to a fuel rich air-to-fuel (A/F) condition in the engine and to the lack or malfunctioning of a emissions control system. Both sampled gasoline and diesel vehicles present a linear (log) correlation between aromatic and CO emission ratios while an anticorrelation is seen between CO and NO_x.

3.2 Vehicle chase experiments

The analytical procedures used for the data obtained with the chase technique can be found in more detail elsewhere (Kolb et al., 2004; Herndon et al., 2005). Briefly, the on-road emissions from a target vehicle are monitored by following it and repeatedly in-

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



tercepting its exhaust emission plumes over a period of several minutes. Similar to the roadside stationary plume sampling, the signals from the emitted species are scaled to the above background exhaust carbon dioxide column concentration signal. The scaling of the above background emitted species to carbon dioxide provides an emission ratio quantifying the ratio of concentrations of the emitted species to the plume excess CO₂ concentrations.

Figure 3 presents the measured on-road mobile emission ratios in the chase sampling mode for both gasoline and diesel vehicles. HDDTs and other large vehicles are intrinsically easier to measure with the chase technique due to the strength of their fresh plume signals and to the ease of intercepting them while directly following the target vehicle. Few LDGVs, which typically emit smaller and cleaner plumes, were targeted with the chase technique. Nevertheless, a number of visibly high emission gasoline-powered vehicles were measured in chase mode, mostly pick-ups and vans, and the results are also included in Fig. 3 for comparison with measurements of emission ratios from HDDT vehicles.

3.3 Fleet average emission ratios

In addition to the chase technique, which focuses on a series of selected individual vehicles within a given vehicular class, fleet average on-road emissions can be obtained by processing randomly intercepted vehicle plumes from surrounding traffic. During the fleet average mode the mobile laboratory measured on-road ambient air mixed with emissions of the surrounding vehicles under various driving modes. As defined in Zavala et al. (2006), we considered driving modes as “Stop and Go” (SAG) for situations when the mobile laboratory was in very heavy traffic conditions, with a vehicle fleet speed of 16 (±8) km/hr for 5 min or more; “Traffic” (TRA) for heavy traffic conditions with a vehicle fleet speed of 40 (±16) km/hr, for 5 or more minutes; and “Cruise” (CRU) for conditions with a moderate to high vehicle fleet speed of 56 m/hr or higher, for 5 min or more. For these experimental settings, it is possible to obtain emission ratios classified by driving mode according to the predominant speed of the traffic. In

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



addition, for measuring emissions under idling conditions (IDL) we used a semi “open-path” approach in which the mobile laboratory drives and samples the emissions along a stationary or semi-stationary line of idling vehicles. The IDL mode measurements were predominantly obtained while sampling the line of vehicles waiting to cross the Mexican-US border between Mexicali and Calexico. In this experiment the mobile laboratory drove on a traffic-free road located beside the border waiting line, entering and re-entering several times, capturing idling and semi-idling emissions from the waiting vehicles.

In the fleet average mode, where even merged plumes from multiple vehicles can be processed and included, the sampling time (and therefore the number of vehicle exhaust plumes intercepted) is normally much larger than for chase mode measurements, providing better statistics. Successful application of this method requires a large sample size of mixed emission periods and long enough sampling times so that the number of sampled vehicles is large enough to include a representative number of high emitters. Care must also be taken to avoid situations where the intercepted plumes are dominated by a few nearby vehicles for significant portions of the sampling period. On the basis of the central limit theorem, the emission averages should then be normally distributed if the samples are unbiased and sufficiently large. In such case, symmetric confidence intervals around the average can be established for fleet emissions estimates. The emission ratios obtained in the fleet average operational sampling mode are appropriate to use for comparison with mobile emissions measured with other high sampling volume techniques and for the validation of an emissions inventory (Zavala et al., 2006).

The measured mobile emission ratios for gases and particle properties sampled in fleet average mode are summarized in Table 1. In all driving modes during the fleet average emissions measurements gasoline vehicles dominated the type of vehicles sampled. Therefore, we will consider these measurements as representative of the gasoline vehicle fleet, not the combined gasoline and diesel fleets. Due to difficulties with inappropriate sampling frequency settings for the AMS instrument, there are

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[◀](#)[▶](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

no available fleet averaged emission PM species partitioning emission ratios for this operational sampling mode.

5 A more detailed view of the effect of driving speed on selected VOCs, NO and CO emissions is shown in Fig. 4. This figure shows the 75th, 50th, and 25th percentiles for selected VOCs, CO and NO emission ratios under the described driving modes. Figure 4 clearly shows that the magnitude and variability of CO and VOCs, which are directly related to the combustion efficiency, are reduced with increasing speed whereas the variability of NO emission ratios does not decrease at higher speeds. C2benzenes and benzene were the highest and lowest abundance aromatic species
10 measured on a mole per mole basis.

Although the number of samples in some driving mode classifications is relatively small, this sampling technique is much more robust for obtaining fleet average emission conditions because their statistics are more significant than those from individual target vehicle emission measurements. Nevertheless, as shown in Table 1, most of
15 the standard deviations are smaller than the observed average. The values reported in Table 1 are used in the following sections for the comparison with similar studies conducted in Mexico City.

4 Discussion

Quantification of emitted fine particles and specific gaseous pollutants often reveals
20 large variability even within a given vehicle type, as indicated by Fig. 2. The measurement of individual plumes in roadside stationary sampling showed that particle number density and NO emission ratios for HDDTs were, in general, higher than those for LDGVs. This is somewhat expected but a larger variability is observed for NO emission ratios from LDGVs. The particularly large variability of NO emission ratios for
25 LDGV may be the result of the different engine combustion temperatures regimes and the lack or malfunctioning of an emissions control system among the sampled vehicles (Wallington et al., 2006). Higher combustion temperatures lead to higher levels of

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



thermal NO_x (Zeldovich mechanism), whereas NO_x from fuel-N varies with the type of fuel.

Results from the vehicle chase measurements in Mexicali (Fig. 3) also show generally higher CO and VOC emission ratios for high emitter gasoline-powered vehicles and high variability for almost all other measured parameters. Aldehydes and other VOCs emission ratios are particularly high for these gasoline powered vehicles, a probable indication of the malfunctioning of, or the lack of, an emissions control system. Similarly, measured NO emission ratios for the high emitter gasoline vehicles are as large as the HDDT emission ratios. However, although the emitted VOCs are higher for the high emitter gasoline-powered vehicles, the organic content in the particle phase of LDGVs still tends to be smaller than for HDDTs. This may partially be due to the higher content of low volatility hydrocarbon molecules in diesel as compared to gasoline fuels and to the better extent of the mixing state of the F/A mixture in gasoline vehicles as compared to diesel (Wallington et al., 2006). Diesel emissions may also contain a larger fraction of unburned motor oil (Canagaratna et al., 2004). Interestingly, the measured variability of the fine particle number density was similar in both types of vehicles but their light absorption, quantifying black carbon content, tends to be smaller for the gasoline vehicles. This is probably also a direct result of the different engine combustion process in the two vehicle classes.

Results from the fleet average measurements showed that NO, fine particle number density and CO emission ratios varied significantly by driving mode whereas the effect is less evident for benzene and practically non-existent for HCHO emission ratios (Fig. 4). These effects of driving modes on emission ratios are consistent with results from Mexico City using the same sampling technique (Zavala et al., 2006). Higher NO emission ratios for higher driving speeds are consistent with higher engine combustion temperatures and higher availability of oxygen in the combustion chamber of gasoline vehicles at those speeds (Kean et al., 2003, Jimenez et al., 1999). Similarly, the production of CO at low vehicle speeds increases as the A/F ratio decreases with less efficient combustion (Cicero-Fernandez, et al., 1997). Whereas NO_x and CO are direct

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[I◀](#)[▶I](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

products of the combustion process, and therefore directly correlated with the A/F ratio or driving speed, the hydrocarbon emissions result from a variety of other processes. These include blowby effects (leakage of gases escaping through sealing surfaces in the engine) during the compression and power strokes, evaporative emissions (whose amount depends on the fuel volatility, temperature and vehicle maintenance) and the combustion process itself. For these reasons, the hydrocarbon emissions result from a mixture of unburned fuel/oil and partially oxidized exhaust products. In general, fuel-based hydrocarbon emissions increase with heavy load conditions and higher power – that is, a rich A/F ratio.

As described above, there are a large number of factors that directly affect the emission characteristics of a given vehicle, all of which affect the observed variability during the sampling of on-road emissions. As such, it is of particular interest to compare the observed variability of the sampled plumes in roadside stationary mode, chase studies and fleet average emission ratios measurements. Comparison of emission ratios obtained in different operational sampling modes provides an opportunity to understand the observed variability of the emission data.

Figure 5 presents the 75th, 50th and 25th percentiles for the fleet average emission ratio measurements as well as all the emission ratios obtained from the chase and the roadside stationary sampling mode measurements of individual gasoline and diesel vehicles plumes. As described above, the four fleet average driving modes in Fig. 5 are more representative of the gasoline vehicle fleet with no significant representation of diesel vehicles. Figure 5 shows that there is higher variability in the sampling of individual plumes and chased vehicles than for the fleet average mode. Also, the variability of the measured emission ratios is larger for the individual plume samples than for the chase events for both gasoline and diesel vehicles. The higher variability of the roadside stationary individual sampled plumes and the chase modes can readily be explained by the “micro” approach of these measurement techniques where a large number of factors (emission control system, vehicle age, maintenance state, fuel type, etc.) may play a major role in determining the emissions from a given vehicle. In

Variability of mobile emissions sampled using a mobile lab**M. Zavala et al.**

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

I◀

▶I

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



the fleet average sampling mode, all these factors are smoothed by averaging (equally weighting) the measured emissions plumes. On the other hand, the variation observed in both the average and the standard deviation in the fleet average sampling mode indicates that the sampling size was large enough to be sensitive to driving mode.

5 Similarly, for emissions of CO, NO and some selected VOCs the resulting consistently smaller standard deviations with respect to the observed emission ratio average in the fleet average mode and the (pronounced in some cases) variation observed with driving mode suggests that the sample size is adequate to represent a true average.

10 The results above indicate that measurements of emission ratios in fleet average mode presented smaller variability than the chasing and stationary operational sampling modes; still, the technique captured the effect of driving conditions on the measured on-road emission ratios. This is of particular importance when using the observed fleet average emission ratios to estimate fuel-based total emission indices for the vehicle fleet which can be used in turn to assess the validity of traditional “bottom up” emissions inventories (Singer and Harley, 2000). The validation of the estimated mobile emission inventories is an important application of the measured mobile emission ratios.

Unfortunately, there was no available information on the local gasoline fuel sales or fuel consumption for Mexicali for the measurement period. However, as a first approximation, we estimated the local gasoline sales by using the readily available national total fuel sales data for Mexico from PEMEX (the national petroleum company of Mexico) and scaled them by the number of vehicles in Mexicali compared to the national values (data which were readily available). This yielded estimated local fuel sales of 1 785 000 liters of gasoline per day for 2005. We focused on estimating fleet average emissions because there was no data available to disaggregate these fuel sales by model year. To that end it is necessary to convert from ppb/ppm of CO₂ emitted to grams per liter of fuel consumed during the combustion process. We assume complete stoichiometric combustion, a typical value of 54.1 moles of carbon per liter of gasoline and a fuel density of 756 grams/liter. This assumption is reasonably valid because the

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



measured emissions levels of exhaust plume CO and VOCs are small compared to the levels of emitted CO₂ (i.e. generally >90% of fuel carbon is emitted as CO₂).

Using these assumptions we estimate the CO and NO_x on-road emissions for Mexicali shown in Table 2. We compare the estimated gasoline mobile emissions in Mexicali with those from the neighbour city of Calexico, CA (these two cities share the Mexico-US border), San Diego, CA, and Mexico City. Mobile emissions for these two Californian cities were obtained from California Air Resources Board (CARB, 2007) for light passenger vehicles whereas mobile emissions from Mexico City were obtained from Metropolitan Environmental Commission (CAM, 2006). Mexicali CO emissions are larger than Calexico by a factor of 8 whereas NO_x emissions are larger by a factor of about 6. These large differences are consistent with the larger fleet size in Mexicali compared to Calexico (a factor of about 9.5). Detailed explanations of the differences between the measurement-based and the model-based emissions estimates are outside the scope of this paper. Nevertheless, an important consideration is that a large number of vehicles cross back-and-forth between the two cities daily, and all are emitting into a shared air basin.

In Table 1, we also compare the gasoline vehicle fleet emission ratios measured during the Mexicali campaign with those obtained in Mexico City during the MCMA-2003 field campaign (Molina et al., 2007). In both campaigns the ARI mobile laboratory obtained the on-road emission measurements using the same techniques, instrumentation and analysis procedures. As such, differences in the reported emission ratios reflect more directly the differences in fleet characteristics and composition between the two urban areas rather than differences in instrumentation, measurement techniques or data analysis procedures. We do not report CO emission ratios for Mexico City because the response time of the CO instrument used during the MCMA-2003 field campaign was not fast enough to fully resolve individual emission plumes. Interesting differences can be found in the data emissions between the two cities. NO emission ratios are ~20% higher in Mexicali than in Mexico City whereas HCHO emission ratios are higher by almost a factor of 2 in Mexico City. However, emission ratios of acetalde-

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



hyde in Mexicali do not seem to be significantly different from those in Mexico City, hence the corresponding HCHO/CH₃CHO ratio varies with the HCHO emission ratio. The elevated HCHO emissions in Mexico City are particularly important since photolysis of HCHO produces HO_x radicals, which initiate tropospheric ozone production and secondary aerosol formation.

Aromatic species emission ratios measured in the Mexicali gasoline vehicle fleet are slightly, but consistently, smaller than those measured in Mexico City. Nevertheless, the variability in the selected VOCs emission ratios seems to be higher in the Mexico City measurements and the difference may not be statistically significant. The variability may be due to the difference in the sampling size (almost a factor of 4 higher in Mexico City) between the two experimental settings. Emission ratios of NH₃ also seem to be higher in Mexico City than in Mexicali by a factor of 2 or more. In general, higher VOC and NH₃ emission ratios are seen in Mexico City possibly due to more prevailing fuel rich conditions induced by Mexico City's much higher altitude and lower ambient oxygen concentration per volume of air.

The variability of the different NO and selected VOCs emission ratios with respect to driving mode seems to be consistent in both datasets although a bit more pronounced in Mexicali than in Mexico City, particularly at cruising speeds. Among the major factors that may play a role in explaining the observed differences between the two measurements are the fleet age, the distribution of vehicle-types, the fraction of vehicles with emission control technology and the fuel composition. For example, using the base year of 1999 for the comparison, the vehicle fleet in Mexicali was on average more than 7 years older and the fraction of vehicles without some emission control technology was about twice that in Mexico City (SEMARNAP, 1999). Other local parameters that may play a role in the differences between the emissions measured in the two urban areas are the temperature, altitude (ambient pressure) and to some extent the relative humidity. In a rapidly growing urban zone such as the US Mexico border, the vehicle fleet size and fuel consumption are continuously changing, effectively making the estimation of mobile emissions a moving target. As mobile emissions clearly play

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[I◀](#)[▶I](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

an important role in the formation of ozone and secondary organic aerosols in urban areas, it would be of major interest to design a follow up study aimed at exploring in detail each of these factors and parameters influencing the differences between the two cities and their correlation with ambient pollution levels.

5 The measurements of on-road NO_y emission ratios in Austin for individual HDDTs sampled in chase mode are presented in Fig. 6 as a function of driving speed. The on-road emission measurements of the trucks, which were identified by their license plates, occurred mostly on an isolated highway at moderate to high speeds. The results indicate a large variability of NO_y emission ratios between individual vehicles as a function of vehicle speed. The observed variability correlated with vehicle speed may also be an indication of enhanced thermal NO_x formation at higher engine temperatures. We compare the diesel NO_y on-road emission ratios from individual HDDTs measured in chase mode in Mexico City, Mexicali and Austin (see Fig. 7). These measurements represent emissions from a limited number of vehicles and it is possible that the sample size is not sufficient to produce fleet average HDDT emission ratios. Nevertheless, within the limitations of the sample size of our data, Fig. 7 indicates that on average NO_y emission ratios from HDDTs in Mexicali and the MCMA were significantly higher than those in Austin and that the variability (indicated here as the 1-sigma standard deviation of the measurements) is similar in all three locations. The large variability observed in the NO_y emission ratios is likely due to the large number of parameters affecting HDDT emissions.

5 Conclusions

We have applied the measurement technique for on-road mobile emission developed in the Mexico City Metropolitan Area during the MCMA-2003 Campaign to Mexicali as part of the Border Ozone Reduction and Air Quality Improvement Program for the Mexicali-Imperial Valley in 2005 and compare similar on-road emission ratios from the two cities. Similar to Mexico City, the measurements in Mexicali were obtained un-

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



der different driving modes representing various speed and congestion characteristics of the fleet and using three different operational sampling modes – roadside stationary sampling, chase studies and fleet average measurements. The analysis focused on the magnitude and variability of the measured emission ratios under the different operational sampling modes.

The observed variability increased from fleet average to chase and roadside stationary sampling for all measured gases and particle emission ratios. The high variability observed in roadside stationary sampling and chase studies can be explained by the large number of factors that can decisively impact the emissions from a given vehicle. The fleet average sampling mode captured the effects of driving conditions on the measured on-road emission ratios. This is important because the measured on-road emission ratios can then be used to estimate fuel-based emission indices used, in turn, to assess the validity of traditional “bottom-up” emissions inventories. Scaling national fuels sales data for Mexicali, we estimated CO and NO_x emissions of 175±62 and 10.4±1.3 metric tons/day, respectively, for the gasoline vehicle fleet. These emissions are 8 and 6 times larger than the emissions estimated for Calexico, CA (the US neighbour border city) due in part to the much larger fleet size in Mexicali.

Comparisons with similarly obtained on-road emissions data in Mexico City indicated that NO emission ratios were around 20% higher in Mexicali than in Mexico City whereas HCHO and NH₃ emission ratios were higher by a factor of 2 in Mexico City. Acetaldehyde emission ratios were not significantly different in the two Mexican cities. Aromatic species emission ratios were similar to or smaller in Mexicali. Differences in reported emission ratios directly reflect the differences in fleet characteristics between the two cities, rather than differences in instrumentation, measurement technique or driving and operational sampling modes. Measurements of NO_y emission ratios from individual chased HDDTs in Austin showed a strong correlation with vehicle speed, similar to the results in Mexicali and Mexico City. However, comparison of the NO_y emission ratios from HDDTs obtained in the three cities, showed that, on average, NO_y emission ratios from HDDTs in Mexicali were higher than in Mexico City, but the ratios

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



from both Mexican cities were higher than in Austin; the variability of the measurements was similar in all three locations.

Acknowledgements. The authors gratefully acknowledge the financial support from the Mexican Metropolitan Environmental Commission (CAM) and the US National Science Foundation (ATM-0528227) for the Mexico City measurements, LASPAU Border Ozone Reduction and Air Quality Improvement Program for the Mexicali measurements, and the University of Texas and the Molina Center for Energy and the Environment for the Austin measurements. Mobile laboratory operational support was provided by Centro Nacional de Investigación y Capacitación Ambiental (CENICA) in Mexico City, Universidad Autónoma de Baja California (UABC) in Mexicali, and the University of Texas in Austin.

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Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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- 15

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Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



Table 1. Comparison of measured fleet-average mobile emission ratios between Mexicali in 2005 and Mexico City in 2003 for various driving modes^a.

	IDL (SD)		SAG (SD)		TRA (SD)		CRU (SD)	
	Mexicali	Mexico City	Mexicali	Mexico City	Mexicali	Mexico City	Mexicali	Mexico City
CO	74.8 (50.5)	N/A	63.7 (29.3)	N/A	64.8 (23.3)	N/A	30.6 (12.5)	N/A
NO	1.28 (0.53)	N/A	5.20 (1.35)	2.92 (0.9)	3.59 (0.45)	4.58 (2.2)	6.50 (1.29)	4.33 (1.7)
HCHO	0.12 (0.04)	N/A	0.18 (0.03)	0.23 (0.06)	0.13 (0.03)	0.23 (0.07)	0.13 (0.03)	0.20 (0.07)
CH ₃ CHO	0.04 (0.01)	N/A	0.06 (0.01)	0.04 (0.02)	0.03 (0.01)	0.04 (0.02)	0.03 (0.02)	0.04 (0.01)
HCHO/CH ₃ CHO	3.1 (0.61)	N/A	3.0 (0.59)	6.2 (1.3)	4.7 (1.54)	6.2 (2.0)	4.5 (1.17)	6.4 (1.8)
Benzene	0.13 (0.09)	N/A	0.10 (0.04)	0.14 (0.04)	0.09 (0.01)	0.10 (0.03)	0.07 (0.02)	0.10 (0.04)
Toluene	0.17 (0.11)	N/A	0.22 (0.09)	0.28 (0.07)	0.17 (0.02)	0.18 (0.06)	0.10 (0.02)	0.18 (0.08)
C2benzenes	0.21 (0.16)	N/A	0.24 (0.09)	0.32 (0.11)	0.19 (0.02)	0.22 (0.09)	0.14 (0.04)	0.19 (0.09)
C3benzenes	0.16 (0.11)	N/A	0.19 (0.10)	0.24 (0.09)	0.13 (0.02)	0.15 (0.05)	0.05 (0.03)	0.15 (0.08)
Aromatics	0.70 (0.47)	N/A	0.79 (0.33)	N/A	0.62 (0.07)	N/A	0.36 (0.05)	N/A
m105	N/A	N/A	0.01 (0.002)	N/A	0.01 (0.002)	N/A	N/A	N/A
m59	0.016 (0.005)	N/A	0.024 (0.011)	N/A	N/A	N/A	0.005 (0.002)	N/A
PND	436 (209)	N/A	N/A	N/A	N/A	N/A	2146 (747)	N/A
PM Absorption	0.031 (0.047)	N/A	N/A	N/A	N/A	N/A	0.069 (0.085)	N/A
NH ₃	0.032 (0.014)	N/A	0.037 (0.006)	0.09 (0.05)	N/A	0.09 (0.06)	0.053 (0.027)	0.11 (0.07)

^a All units are in ppb/ppm-CO₂ except for particle number density (PND) [part/cc/ppm-CO₂] and PM light absorption [Mm⁻¹/ppm-CO₂] and HCHO/CH₃CHO [ppb/ppb]. We consider here C2Benzene as the sum of xylene isomers, ethylbenzene, and benzaldehyde and C3Benzene as the sum of C₉H₁₂ isomers and C₈H₈O isomers. Aromatics are the sum of benzene, toluene, C3benzene and C2benzene. m105 and m59 generally refer to ion masses m/z signals that are related to ethenylbenzene and acetone, respectively. IDL: idle; SAG: Stop and Go; TRA: Traffic; CRU: Cruise conditions. N/A: Not available, SD: 1 standard deviation. See text for details.

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

Table 2. Comparisons of gasoline fleet mobile emissions [tons/day] in Mexicali (this study) with those estimated for other urban areas. (See comments in text).

Pollutant	Mexicali	Calexico, CA ^a	San Diego, CA ^a	Mexico City ^b
CO	175±62	21.8	244.7	2765
NO _x	10.4±1.3	1.9	21.1	188

^a Data from CARB, (2007) for the 2006 light duty passanger vehicles emissions.

^b Data from the 2004 MCMA emissions inventory (CAM, 2006) for LDGVs.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

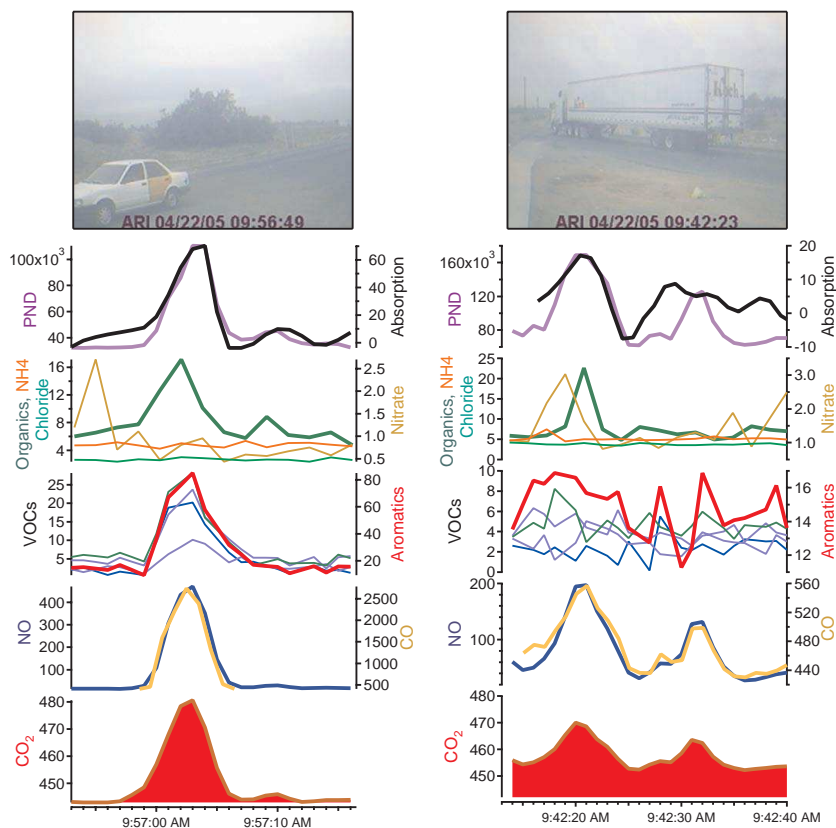


Fig. 1. Roadside stationary exhaust emission measurements of a LDGV (left panels) and a HDDT (right panels). All pollutant concentration units are ppbv, except for CO₂ [ppmv], AMS fine PM non-refractory composition [$\mu\text{g}/\text{m}^3$], PM light absorption [Mm^{-1}] and fine particle number density (PND) [$\text{particles}/\text{cm}^3$].

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



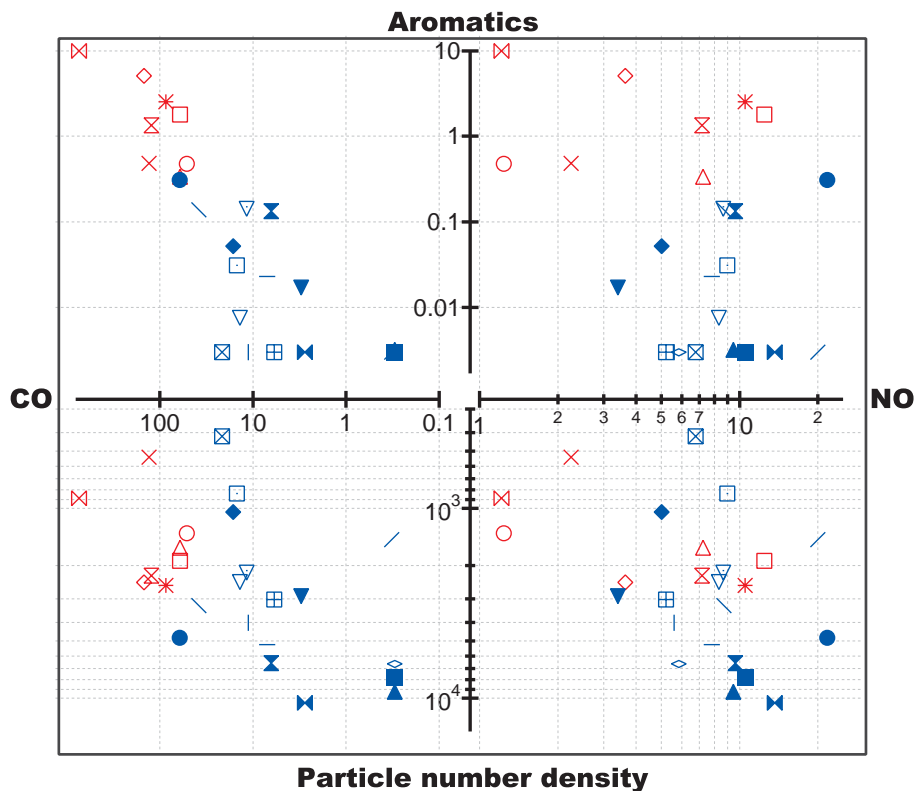
Variability of mobile emissions sampled using a mobile lab**M. Zavala et al.**

Fig. 2. Comparison of emission ratios for CO [ppb/ppm- CO_2], NO [ppb/ppm], particle number density (PND) [particles/ cm^3 /ppm], and aromatics [ppb/ppm] (sum of benzene, toluene, C3-benzenes and C2-benzenes) of individual vehicles sampled in stationary mode for gasoline (red) and diesel (blue) vehicles.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

I◀

▶I

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

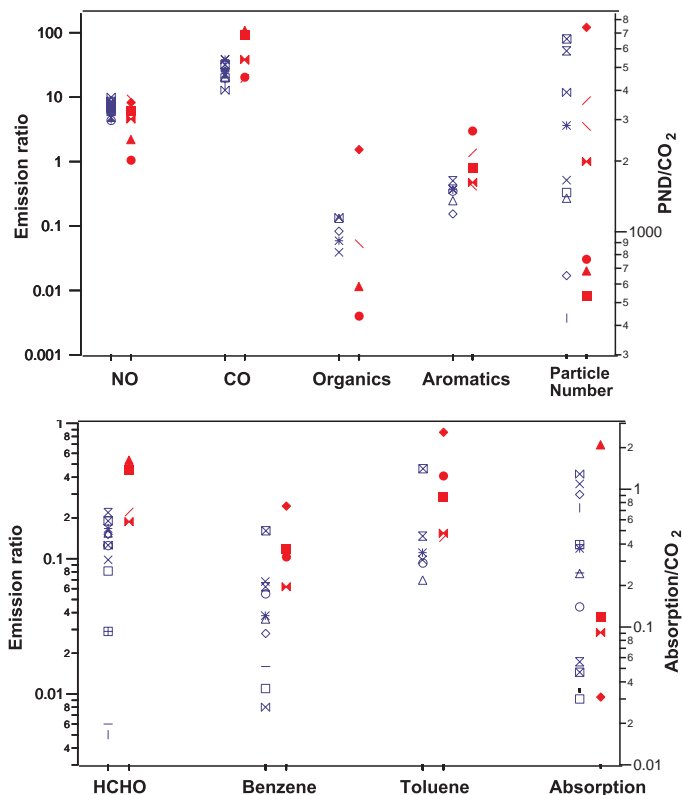


Fig. 3. Mobile emission ratios measured from individual gasoline (red) and diesel (blue) vehicles. Each symbol represents an individual chaser vehicle. “Aromatics” refers to the sum of benzene, toluene, C3-benzenes and C2-benzenes. “Particle number” [particles/cm³/ppm] refers to particle number density (PND). “Organics” refers to the organic component of the fine aerosol mass [$\mu\text{g}/\text{m}^3/\text{ppm}$] less than 1 μm in diameter. “Absorption” refers to PM light absorption [$\text{Mm}^{-1}/\text{ppm}$]. All other units are in [ppb/ppm].

[Title Page](#)
[Abstract](#)
[Introduction](#)
[Conclusions](#)
[References](#)
[Tables](#)
[Figures](#)
[◀](#)
[▶](#)
[◀](#)
[▶](#)
[Back](#)
[Close](#)
[Full Screen / Esc](#)
[Printer-friendly Version](#)
[Interactive Discussion](#)

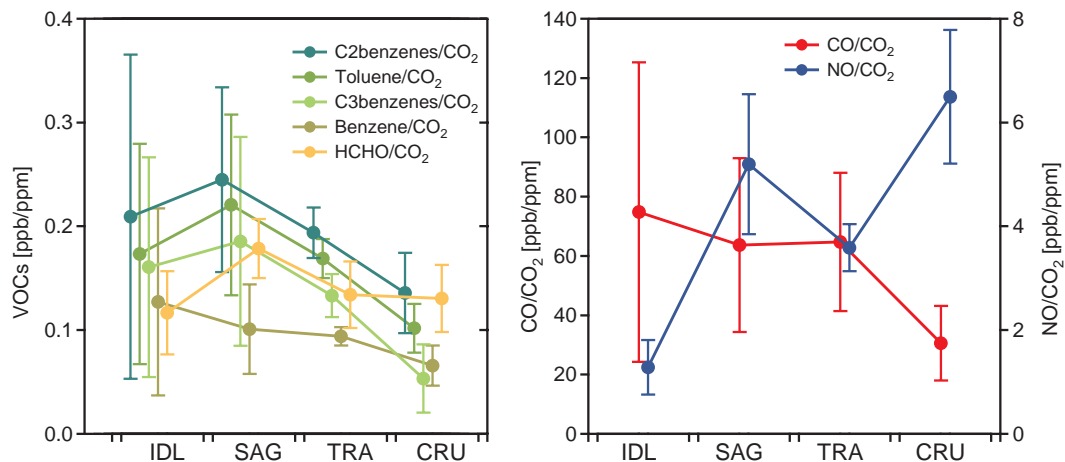

Variability of mobile emissions sampled using a mobile lab**M. Zavala et al.**

Fig. 4. Fleet average mobile emission ratios by driving mode for CO, NO and selected VOCs showing the 75th, 50th, and 25th percentile for each driving mode.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[◀](#)[▶](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

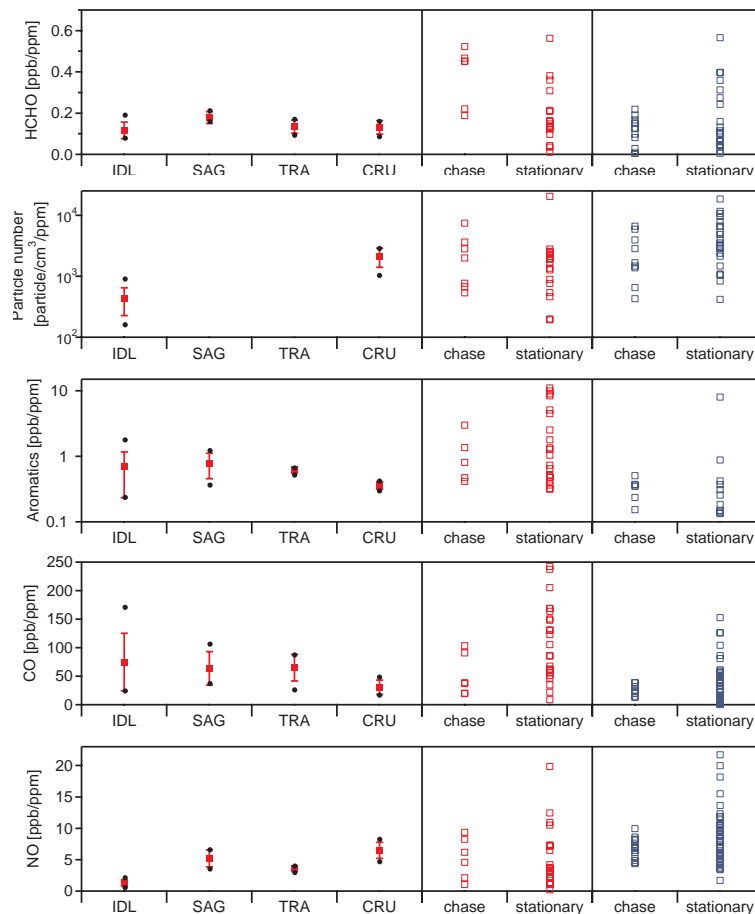
Variability of mobile emissions sampled using a mobile lab**M. Zavala et al.**

Fig. 5. Comparison of measured on-road mobile emission ratios for various pollutants by sampling operational for gasoline (red) and diesel (blue) vehicles. “Aromatics” refers to the sum of benzene, toluene, C3-benzenes and C2-benzenes.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[◀](#)[▶](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

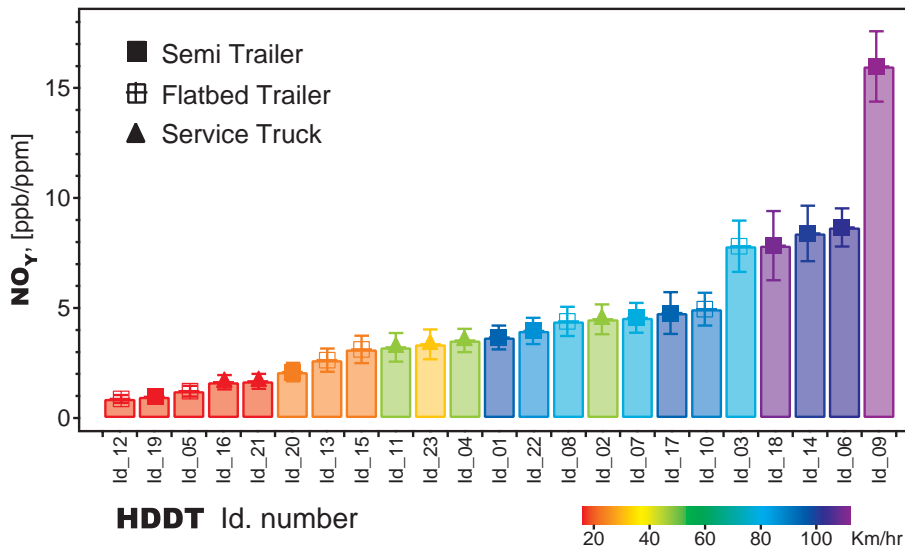


Fig. 6. NO_y/CO_2 on-road emissions of individual HDDTs measured in Austin TX, by vehicle speed.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



Variability of mobile emissions sampled using a mobile lab

M. Zavala et al.

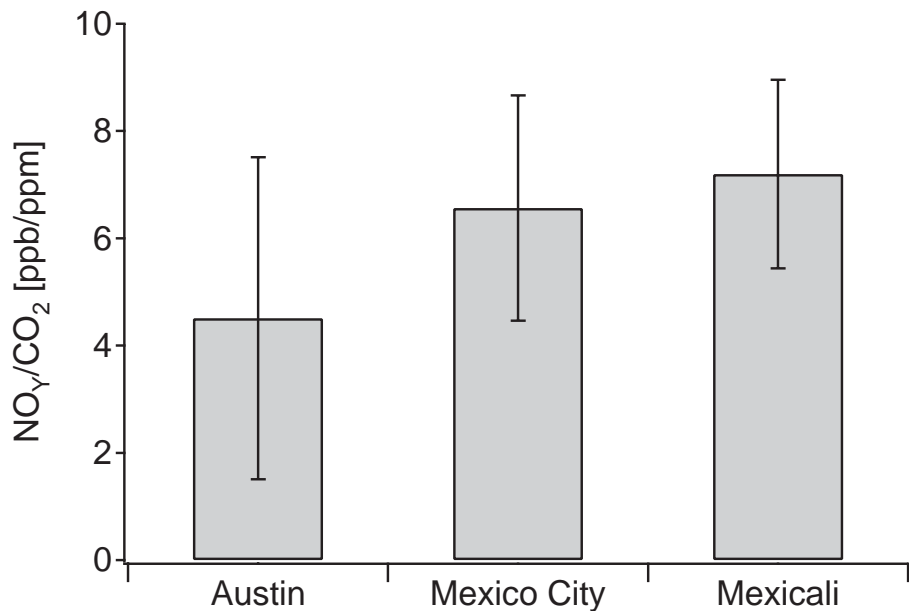


Fig. 7. Comparison of on-road NO_y/CO_2 emission ratios measured in Austin, Mexicali and Mexico City for HDDTs. The error bars represent the 1 standard deviation of the data.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

